

Chapter 4: Smart (And Not So Smart) Mixes of New Environmental Policy Instruments

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Abstract

The chapter analyses the adoption and deployment of traditional ‘command-and-control’ regulations and ‘new’ environmental policy instruments (NEPIs) as they occur in practice in five different jurisdictions, namely Austria, Germany, the Netherlands and United Kingdom (UK) as well as the European Union (EU) since the early 1970s. It focuses on three different types of NEPIs - informational instruments, voluntary agreements and market-based instruments – and examines how and why they have become mixed in different jurisdictions. It argues that whilst there has been a significant uptake of NEPIs in all five jurisdictions, important differences have remained as regards the composition of instrument mixes in particular jurisdictions. Adopting a longitudinal perspective allows for the identification of leaders, followers and laggards for different types of NEPIs. Although there may be a theoretical ‘optimal mix’ of policy instruments, in reality patterns of adoption and deployment are very strongly influenced by a mixture of contingent factors which vary within and across jurisdictions, over time.

Keywords

New environmental policy instruments (NEPIs), traditional regulation, smart instrument mixes, retooling, government, governance, European jurisdictions

1. Introduction

When environmental policy emerged as a distinctive policy field in the early 1970s, traditional ‘command-and-control’ regulation quickly became the dominant environmental policy instrument in most European jurisdictions and in North America (e.g. Faure, Vervaele and Weale 1994; Bemelmans-Videc, Rist and Vedung 1998; Gunningham and Grabovsky 1998; Wurzel, Zito and Jordan 2013). The United Kingdom’s (UK) initial reluctance to adopt statutory environmental laws constituted an exception (e.g. Vogel 1986) which, due to domestic reasons (e.g. the privatisation of the water industry necessitated the more legalistic approach) and external reasons (e.g. the increasing Europeanisation of British environmental policy), largely came to an end in the late 1980s (e.g. Jordan 2002; see also Table 4.1 below).

Gunningham and Grabovsky’s (1998: 18) statement that ‘in Europe, there remains heavy reliance on command-and-control as the basic instrument and very limited experimentation and policy mixes’ still remains broadly correct although, for reasons which will be explained in this chapter, it applies more strongly to some European countries than others and is relevant more in particular time periods compared to others. In recent years most European countries have adopted an increasingly wider range of ‘new’ environmental policy instruments (NEPIs) without however abandoning traditional regulatory tools. This has produced often complex mixes of ‘new’ and ‘old’ instruments that exist side by side or, in some cases, have mutated into hybrid instruments (e.g. Gunningham and Grabovsky 1998; Jordan, Wurzel and Zito 2005; Wurzel, Zito and Jordan 2013). The need to orchestrate better such complex policy instrument mixes has been further exacerbated by the rise of international environmental agreements which have also stipulated new instruments or new variants of old instruments (Abbott 2012). As ‘old’ and ‘new’ policy instruments have increasingly been combined either by design or default, questions about ‘smart mixes’ (Gunningham and Grabovsky’s 1998) and ‘not so smart mixes’ (e.g. Howlett and Rayner 2004) have become more pertinent. Importantly, the question about instrument mixes

‘has become a very central question’ not only for academics but also for practitioners (Interview, German Environment Agency official, 2017).

The debate about ‘optimal’ policy instruments and/or instrument mixes was initially dominated by economists who usually advocated the use of market-based instruments over traditional ‘command-and-control’ regulation on efficiency grounds (e.g. Siebert 1976; Baumol and Oates 1979; Burrows 1979). Economists often either ignored or downplayed the significance of contextual factors (such as national regulatory traditions and party political preferences) for policy instrument selection although there are important exceptions (e.g. OECD 1994). Legal scholars and political scientists, on the other hand, have typically emphasised the importance of legal and political context variables (such as their compatibility with particular legal systems and the politics of instrument selection) in addition to effectiveness and efficiency criteria. Majone (1976) and Hood (1983: 9) put forward some of the first comprehensive policy instrument studies from a political science perspective in which they argued that policy instrument choice is an inherently political process. As Hood (1983: 136) has argued: ‘very commonly it is the instrument selected for reaching a policy aim that is far more contentious than the aim itself’. Seen from a more political perspective, policy instrument choice is not only about solving environmental problems efficiently and effectively but also about who gains and who loses from the adoption of which type of policy instrument (e.g. Hood 1983; Lascoumes and Le Gales 2007; Wurzel, Zito and Jordan 2013). It is therefore unsurprising that the selection of policy instruments and their exact design features, which may vary considerably from jurisdiction to jurisdiction, often attracts considerable lobbying activities from non-state actors who will be affected by these instruments.

In order to more precisely assess how different instruments interact to create mixes, it is important to assess the characteristics of the instrument types found in the policy sector. Consequently, this chapter focuses on the use of traditional ‘command-and-control’ regulation and three different types of NEPIs - informational instruments, voluntary agreements and market-based instruments – in five different jurisdictions, namely Austria, Germany, the Netherlands and United Kingdom (UK) as well as the European Union (EU) since the early 1970s. As informational instruments are widely perceived as relatively ‘soft’ instruments and traditional ‘command-and-control’ regulations as relatively ‘hard’ instruments with voluntary agreements and market-based instruments falling somewhere in between, this chapter covers the full spectrum which ranges from ‘soft’ to ‘hard’ instruments.

2. Government and governance

Before assessing in more detail the adoption and usage patterns of ‘new’ and ‘old’ environmental policy instruments across jurisdictions and time, we locate the examination about ‘smart’ and ‘not so smart’ mixes of policy instruments in the wider debate about *government* and *governance*. Governance and government can be perceived as different forms of *governing* (Finer 1970). If the ‘strong state’ is characterized by the extreme form of top-down government in the era of ‘big government’ (e.g. Kitschelt et al. 2003), then the equally extreme form of bottom-up governance is represented by self-organising societal systems which are almost insusceptible to steering efforts by governments (e.g. Luhmann 1984). These two extreme positions can rarely be found in highly developed (European) liberal democracies although they constitute useful heuristic analytical devices which can help us to understand better the central differences between traditional tools of government (i.e. top-down ‘command-and-control’ regulation) and new modes of governance (such as NEPIs and other new modes of governance) as well as the policy instrument selection process. In the international relations arena somewhat

similar extreme positions have been identified in the form of top-down multilateral agreements which stipulate legally binding targets and timetables and bottom-up polycentric governance arrangements which rely on voluntary pledges and a high degree of self-coordination.

It is widely accepted that NEPIs can be used as yardsticks to assess empirically whether less hierarchical new modes of *governance* have supplanted or merely supplemented top-down ‘command-and-control’ regulation (e.g. Bemelmans-Videc, Rist and Vedung 1998; Jordan, Wurzel and Zito 2005; Holzinger, Knill and Schäfer 2006; Héritier and Lehmkuhl 2008; Wurzel, Zito and Jordan 2013). While much of the general *governance* literature has claimed that a decisive shift from traditional top-down state-led *government* towards bottom-up societal *governance* has taken place (e.g. Rhodes 1996; Stoker 1998), most of the more empirically informed NEPIs studies have argued that ‘new’ policy instruments have either supplemented (rather than supplanted) traditional regulation or have formed new hybrid policy instruments with features from both ‘old’ and ‘new’ instruments (e.g. Gunningham and Grabosky 1998; Wurzel, Zito and Jordan 2013).

3. ‘New’ environmental policy instruments

Much of the policy instrument literature distinguishes between traditional ‘command-and-control’ regulatory instruments and ‘new’ modes of environmental governance or NEPIs (Gunningham and Grabosky 1998; Holzinger and Knill 2003; Jordan, Wurzel and Zito 2005; Wurzel, Zito and Jordan 2013). However, the definition of what constitutes a ‘new’ environmental policy instrument needs to be established empirically. While certain NEPIs are new in some jurisdictions they may have already been in use in other jurisdictions for some time. For example, Germany adopted the world’s first eco-label scheme in 1978 while Austria, the

Netherlands and the EU adopted somewhat similar schemes only in 1992 (Jordan et al. 2004; Wurzel, Zito and Jordan 2013).

There is no universally accepted definition of which instrument types constitute NEPIs although the following three-fold typology is relatively widely accepted (e.g. Bemelmans-Videc, Rist and Vedung's 1997; Bähr 2010; Jordan, Wurzel and Zito 2005; Wurzel, Zito and Jordan 2013): (1) informational instruments (which are sometimes also termed suasive instruments), (2) voluntary agreements, and (3) market-based instruments. Our relatively parsimonious threefold typology has the analytical advantage of making it easier to identify shifting patterns in instrument use and thus also policy instrument mixes across jurisdictions and time than would be possible with a more complex typology.

3.1. Informational instruments

Eco-label schemes and environmental management schemes (EMSs) constitute important examples of informational policy instruments. They are both relatively 'soft' instruments which rely on voluntary participation while providing participants with means to publicise their relatively good environmental performance (e.g. Wurzel, Zito and Jordan 2013).

Corporate actors which operate in markets with a high level of public environmental awareness and/or 'green consumerism' sometimes apply for eco-label schemes or EMSs also to avoid a competitive disadvantage vis-à-vis competitors which already participate in such schemes. The levels of public recognition of eco-label schemes and EMS as well as the participation rates are important factors for determining the success or failure of these two sub-types of informational instruments. Similar pressures apply to voluntary international certification and labelling schemes.

3.1.1 Eco-labels

Informational policy instruments such as eco-label schemes are frequently also labelled ‘moral suasion’ instruments because they provide citizens and consumers with standardized information about the environmental impact of certain products and services with the aim of encouraging them to adopt more sustainable purchasing decisions (e.g. Jordan *et al.* 2004; Wurzel, Zito and Jordan 2013). The OECD (1991) differentiates between the following three sub-types of eco-label schemes: Type I – externally verified, multi-issue schemes; Type II – unverified self-declaratory schemes by manufacturers and/or retailers; and Type III – single issue schemes which are based on quantified product information based on life-cycle impacts (e.g. the product profile of a particular product model). Type I eco-label schemes, which were first designed in Europe (i.e. in Germany) where they have become widely used although their popularity varies considerably between different European jurisdictions. Type II schemes are widely used by corporate actors across the world although they tend to have a poor reputation in terms of their environmental ambition and implementation (Jordan *et al.* 2004; OECD 1997). Type III eco-label schemes are usually narrowly focused (e.g. on forestry products) and not (yet) very widely used in Europe. For space constraints and because they are most widely used in Europe, this chapter focuses only on Type I eco-label schemes.

3.1.2 Environmental Management schemes

Environmental management systems such as the EU’s Environmental Management and Audit Scheme (EMAS) and the International Standard Organisation’s (ISO) 14001 standard, provide incentives for industry to adopt measures for achieving certain environmental objectives. Both EMAS and the ISO 14001 require environmental impact audits, the setting up of internal management systems with the aim of reducing the negative environmental impact and the

publication of regular public statements. Companies (in the case of EMAS and ISO 14001) and public organisations (in the case of EMAS) which achieve the required standards are granted an official confirmation and permitted to use a logo for marketing purposes. Although the EU's EMAS and the ISO 14001 are voluntary instruments, public and/or market pressure may push companies to participate for fear of losing out to competitors. Several European governments (e.g. Austria and Germany) have linked EMAS participation to lighter regulatory regimes (e.g. fewer inspections) thus creating additional incentives for this scheme.

3.2 Voluntary Agreements

There is no universally accepted definition of voluntary agreements. While the EU Commission defines voluntary agreements as 'agreements between industry and public authorities on the achievement of environmental objectives' (CEC 1996: 5) the European Environmental Agency (EEA) has adopted a narrower definition which covers 'only those commitments undertaken by firms and sector associations, which are the result of negotiations with public authorities and/or explicitly recognised by the authorities' (EEA 1997: 11).

Börkey and Lévêque's (1998) have differentiated further voluntary agreements into: (1) unilateral commitments, (2) negotiated agreements, and (3) public voluntary schemes. *Unilateral commitments* are self-declaratory improvement measures put forward by corporate actors. *Negotiated agreements* are more formal agreements between industry and public authorities. Many negotiated agreements in the Netherlands are 'covenants' which are legally binding although in practice it can be difficult to enforce them through the courts. In Austria and Germany voluntary agreements cannot be legally binding for constitutional reasons. Austria and Germany could therefore not have accepted legally binding voluntary agreements which the European Commission considered as a possible policy instrument on the EU level in the early

2000s (CEC 2002; Wurzel, Zito and Jordan 2013). Negotiated agreements are arguably closer to the government end on the government-governance dimension than unilateral commitments which are closer to the governance end. This applies in particular to the Dutch covenants which Gunningham and Grabosky (1998: 40) have described as ‘innovative version of command and control’. *Public voluntary schemes*, which are not widely used in Europe, are established by public bodies. Individual companies are free to join such schemes, although public authorities define the membership criteria. This chapter focuses only on negotiated agreements for space constraints and because they are the most widely used sub-type voluntary agreements in Europe.

3.3 Market-Based Instruments

Eco-taxes and emissions trading schemes (ETSs) are widely seen as the most important market-based policy instruments. The OECD (1994: 17) defines market-based instruments as tools which affect ‘estimates of costs of alternative actions open to economic agents’. It distinguishes between the following four main types of market-based instruments: (1) eco-taxes (including charges and levies); (2) emissions trading; (3) subsidies; and, (4) deposit-refund schemes. In Northern Europe eco-taxes have long been widely used. Emissions trading schemes were pioneered in the United States of America (USA) on a regional basis (for sulphur dioxide emissions) already in the 1980s while European jurisdictions started to experiment with them (for carbon dioxide emissions) only in the early 2000s (e.g. Ellerman, Buchner and Carraro 2007; Wurzel 2008). However, in 2003 the EU adopted the world’s first transnational/supranational ETS which became operational in 2005.

Subsidies (e.g. for renewable energy) and fiscal incentives (e.g. for less polluting cars) are relatively widely used in Europe. Although there are important exceptions (e.g. OECD 1999), most neoliberal economists have been critical of subsidies and fiscal incentive for less

environmentally damaging activities because they consider them to constitute market-distorting instruments. Subsidies for environmentally damaging activities (e.g. energy production in subsidised coal-fired power stations) have long been criticized by economists and environmental groups as well as environment agency officials (Interviews, 2017) although governments have found it difficult to abolish them. The empirical focus of this chapter will be on eco-taxes and emissions trading schemes which are widely seen as the most important market-based environmental instruments in Europe.

4. Adoption Patterns of NEPIs and Regulation in Different Jurisdictions Over Time

Before assessing in more detail the adoption and usage patterns of particular types of ‘new’ and ‘old’ policy instruments, we first analyse the general overall instrument adoption patterns across jurisdictions and over time. This will allow for both the identification of leaders, followers and laggards for particular policy instruments and the detection of jurisdictional policy instrument mixes and how they have changed over time. As explained in Chapter 1 by van Erp et al., policy mixes are rarely designed as a totality; instead a more gradual layering process occurs as policy-makers experiment with new ideas and instruments in their particular jurisdictional context.

Table 4.1 summarizes the adoption rates of the above mentioned three types of NEPIs and environmental regulation across five European jurisdictions between the 1970s and 2010s. Table 4.1 uses simple weighting categories – low, medium and high - to allow for meaningful comparisons between four different instrument types across five jurisdictions over a period of almost 50 years. The data in Table 4.1 is largely based on a study by Wurzel, Zito and Jordan (2013), who assessed a wide range of primary documents and undertook more than 50 interviews in all four member states as well as on the EU level between 2002-2012. It has been updated with additional primary research (including interviews) in 2017. Importantly, the descriptors

used in Table 4.1 offer a relative comparison across jurisdictions and time, rather than an absolute baseline judgement.

Table 4.1: The use of different NEPIs and environmental regulations since the 1970s

	Jurisdiction	1970s	1980s	1990s	2000s	2010s
Instrument type						
Eco-label scheme	Austria (*1990)	---	---	Medium	Medium	Medium
	Germany (*1978)	Medium	High	High	High/medium	Medium
	Netherlands (*1992)	--	---	Low	Low	Low
	UK	---	---	---	---	---
	EU eco-label (*1992)	---	---	Low	Low	Low
EU EMAS (*1993)	Austria	---	---	High	High/Medium	High/medium
	Germany	---	---	High	High/medium	High/medium
	Netherlands	---	---	Low	Low	Low
	UK	---	---	Low	Low	Low

	EU average	---	---	Low	Low	Low
Voluntary Agree- ments	Austria	Low	Low	Medium/ high	Medium/ Low	Low
	Germany	Low	High	High	Medium/ Low	Low
	Netherlands	Low	High	High	Medium	Low/ medium
	UK	Low	Low/ medium	Low/ medium	Low	Low
	EU-wide use	---	---	Low	Low	Low
Eco-taxes	Austria	Low	Medium	Medium	Low	Low
	Germany	Low	Medium	High	Medium	Low
	Netherlands	Medium	High	High	Medium	Low
	UK	Low	Low	Medium	Medium	Low
	EU-wide use	---	---	---	---	---
ETS	Austrian ETS	---	---	---	Low. EU ETS in 2005	EU ETS
	German ETS	---	---	---	Low. EU ETS in 2005	EU ETS
	Dutch ETS	---	---	Low	Medium national experiments.	EU ETS

					EU ETS in 2005	
	UK ETS (*2002)	---	---	---	High/Medium. National ETS replaced by EU ETS in 2005	EU ETS
	EU ETS (*2005)	---	---	---	High/medium	High/medium
Environmental regulation	Austria	Low	Medium	High	Medium	Medium
	Germany	High	High	High	High	Medium
	Netherlands	High	High	High/medium	Medium	Medium
	UK	Low	Medium	Medium	Medium	Low
	EU	High	High	High	Medium	Medium/low

Note: () Refers to the year in which a particular instrument was adopted first.*

Source: Adapted from Wurzel, Zito and Jordan (2013: 213-214) with updates based on interviews carried out in 2017.

The following broad four conclusions can be drawn from Table 4.1. First, the use of NEPIs has overall become more widespread since the 1970s. All five jurisdictions have adopted some NEPIs. This finding is in line with the existing policy instrument literature which has argued that

there has been a significant rise in the adoption of NEPIs across a wide range of jurisdictions (e.g. Bemelmans-Videc, Rist and Vedung 1997; De Bruijn and Hufen 1998; Holzinger and Knill 2003; Bähr 2010; Jordan, Wurzel and Zito 2005; Wurzel, Zito and Jordan 2013). Gunningham and Grabosky's (1998: 89) statement that 'environmental policy is in transition: from command and control towards a much more pluralistic conception of instrument design' therefore still seems largely applicable although there has been a decline in the use of certain NEPIs. For example, voluntary agreements have been less popular in the 2000/10s than they were in the 1980s/90s.

Second, although there has been an overall increase in the use of NEPIs it has taken place at various speeds across different jurisdictions. Considerable fluctuations have occurred in the adoption patterns of particular NEPIs (as well as traditional regulations) across different jurisdictions (see also Wurzel, Zito and Jordan 2013). For example, as will be explained in more detail below, the Netherlands and Germany have adopted a much larger number of voluntary agreements than Austria, the EU and UK. While the UK has remained the only jurisdiction which has failed to adopt its own eco-label scheme it acted as an emissions trading leader when it set up a national pilot scheme before the EU ETS became operational in 2005.

Third, although NEPIs are now overall more widely used in all five jurisdictions compared to the early 1970s, they can be more commonly found in some jurisdictions (e.g. the Netherlands and Germany) than in others (e.g. the EU). Some jurisdictions have been early adopters of particular types of NEPIs but ambivalent about or even hostile towards other types of NEPIs. For example, Germany adopted its eco-label scheme in 1978 while Austria and the Netherlands as well as the EU adopted their own eco-label schemes only in 1992. Austria and Germany did

not have in place emissions trading schemes with which the Netherlands and in particular the UK experimented much earlier on (Wurzel, Zito and Jordan 2013).

Fourth, the adoption patterns of NEPIs (and traditional regulation) can vary not only across jurisdictions but also across time. For example, the Netherlands and Germany both adopted more voluntary agreements in the 1980s/90s than in the 2000s/10s. The adoption of the EU ETS constitutes one important factor for the decline in Dutch and German voluntary agreements which were particularly widely used for climate change issues. In these cases one type of NEPI (i.e. emissions trading) seems to have at least partly supplanted another type of NEPI (i.e. voluntary agreements). This empirical finding is in line with Hood's (1983: 126) argument that a 're-tooling' process may take place from time to time.

Salomon (2002: 18) has pointed out that '[f]ar from simplifying the task of public policy solving, the proliferation of tools has importantly complicated it even while enlarging the range of options and the pool of resources potentially brought to bear'. Questions have therefore been raised about what constitutes an 'optimal' or 'smart' mix of environmental policy instrument (e.g. Gunningham and Grabosky 1998; Jordan, Wurzel and Zito 2007).

As will be explained in more detail below, what complicates any cross-jurisdictional comparisons, is the fact that the same type of NEPI may be used in dissimilar ways in different jurisdictions (e.g. Jordan, Wurzel and Zito 2005; Wurzel, Zito and Jordan 2013). When comparing the use of particular policy instruments and/or instrument mixes is therefore important to take into account how exactly a certain NEPI is used in a particular jurisdiction.

The chronological breakdown of NEPIs usage in Table 4.1 above enables us to identify patterns of policy instrument sequencing. Much of the existing policy instrument literature advocates a sequencing which starts with softer (i.e. less coercive) instruments before the adoption of harder (i.e. more coercive) instruments (e.g. Gunningham and Grabosky 1998; Salamon 2002; Vedung 1997). While drawing on Doern and Wilson (1974), Vedung (1997: 40) has argued that

‘the least coercive instruments are introduced first in order to gradually weaken the resistance of certain groups of individuals and adjust them to government intervention in the area. After some time, the authorities feel entitled to regulate the matters definitely by employing their most powerful instrument’.

Similarly, Gunningham and Grabosky (1998: 123) have argued for the sequencing of environmental policy instruments: it enables ‘escalation from the preferred least interventionist option, if it fails, to increasingly more interventionist alternatives’.

The gradual ratcheting upwards of policy instruments according to their degree of coerciveness - starting with soft policy instruments (i.e. horizontal self-governance) and ending with hard policy instruments (i.e. coercive top-down government) - is principally also in line with liberal/neoliberal economic thinking. However, the empirical evidence put forward in this chapter shows a different pattern (see also Wurzel, Zito and Jordan 2013). Germany, the Netherlands and EU all relied almost exclusively on traditional environmental regulations in the 1970s. Austrian and in particular UK environmental policy initially relied less heavily on environmental regulations. However, by the 1980s, environmental regulations were also the dominant policy instrument in Austria. In 1970s and 1980s, EU environmental policy relied almost exclusively on traditional environmental regulation (in the form of directives, regulations and decisions). Importantly, in four out of five jurisdictions traditional environmental regulations were adopted first before they were supplemented by softer environmental policy instruments.

Germany's early and heavy reliance on traditional environmental regulation instead of NEPIs can be explained with reference to its dominant policy style and its state of law (*Rechtsstaat*) tradition as well as the fact that voluntary agreements or other soft policy instruments were 'deemed inappropriate for the defence against dangers (*Gefahrenabwehr*) to the environment and human health. You need regulations for such a task' (Interview, German Environment Ministry official, 2001). The UK was initially reluctant to adopt statutory environmental standards. However, this gradually changed when it had to implement a rapidly increasing number of EU environmental regulations from the mid-1970s onwards. Around that time EU environmental legislation therefore became a major driver for UK environmental policy. Clearly, particular policy instruments are considered more appropriate and legitimate in certain jurisdictions regardless of their theoretical advantages, for example, in terms of efficiency (e.g. Howlett 1991; Salamon 2002: 24).

4.1 Usage Patterns of Informational Instruments

4.1.1 Eco-labels

Germany adopted the world's first national eco-label scheme in 1978. Austria (1991) and the Netherlands (1992) followed with their own national eco-label schemes. In 1992 the EU adopted an EU-wide eco-label scheme which supplements (rather than supplants) member states' national eco-label schemes. The UK (is one of several EU member states which) has relied solely on the EU eco-label. However, the EU eco-label scheme continues to suffer from relatively low participation rates which are at least partly due to low recognition rates among the general public who tend to know more about their national eco-label scheme where such a scheme exists. However, the popularity of national eco-label schemes has also undergone certain ups and downs over time. Table 4.1 illustrates that the German Blue Angel eco-label scheme was significantly more popular in the 1980s/90s than in the 2010s.

During the early 1990s, a rapid global diffusion of eco-labels took place although not all eco-label schemes became successful (Jordan et al. 2004). The Dutch and, although to a lesser degree, the Austrian eco-label schemes still suffer from relatively low take up while some countries (e.g. the UK) decided against the adoption of national eco-label schemes. The German Blue Angel scheme acted as a catalyst, but eco-label scheme followers did not simply copy it. For example, the Austrian, Dutch and EU eco-label schemes have put more emphasis on sophisticated life-cycle analysis while the German eco-label scheme has relied more strongly on simplicity with the aim of communicating clear messages to consumers and participants. Importantly, the various national eco-label schemes reflect national preferences: the German scheme puts a lot of emphasis on products (and services), Austria pioneered a tourism eco-label and the Dutch were first to include the food and flower sectors.

4.1.2 Environmental Management Schemes

In 1993 the EU adopted EMAS which became applicable in 1995. The EU negotiated EMAS at about the same time as the ISO discussed the 14001 standard which created a similar, although less demanding, voluntary EMS which has worldwide recognition. The UK, which had already in place national standards for domestic EMSs, initially supported both the EU's EMAS and the ISO 14001. However, while the ISO 14001, which closely resembled the British national EMS standards, has achieved a relatively high adoption rate in the UK, the EU's EMAS has not done so partly because its requirements significantly exceeded the British Standard Institution's/ISO's standards. Austria and Germany were initially highly sceptical about EMAS for the following two main reasons. First, EMAS constitutes a procedural policy instrument which initially did not fit easily with the dominant Austrian and German environmental regulatory styles that are both characterised by a high number of substantive policy measures which are derived from the best

available technology (BAT) (*Stand der Technik*) principle. Second, Austrian and German environmental policy makers were initially weary that self-regulatory environmental management systems (in the form of EMAS and ISO 14001) might weaken substantive environmental standards. However, by the 2000s, about 70% of all registered EMAS sites were in Austria and Germany. One reason for the surprising popularity of EMAS in Germany is due to the fact that the German government has combined EMAS with traditional regulatory instruments (e.g. by reducing the environmental regulatory burden in the form of fewer inspections for companies which successfully participate in EMAS).

4.2 Usage Patterns of Voluntary Agreements

The adoption of voluntary agreements has overall grown significantly since the 1970s despite concerns about their effectiveness (OECD 2003). As was pointed out already above, the most popular type of voluntary agreement within the EU is the negotiated agreement. However, even for this particular sub-type of voluntary agreements significant jurisdictional differences exist in terms of their specific design and usage. Since the 1990s, most Dutch voluntary agreements have taken the form of legally binding covenants (e.g. Mol, Lauber and Liefferink 2000). The Dutch covenants, which are negotiated between industry and government within a fairly formalised process, stipulate monitoring requirements and are at least in theory enforceable through the courts. All German voluntary agreements are non-binding, although they are often negotiated in the ‘shadow of the law’ and thus often proposed by industry as a means of pre-empting regulation (Wurzel, Zito and Jordan 2013). Compared to the Dutch covenants Austrian and German voluntary agreements are developed in a much more informal manner and sometimes even fail to stipulate explicit monitoring requirements although notable exceptions exist. In the UK relatively few (negotiated) voluntary agreements exist most of which are non-binding and very flexible.

The uptake of voluntary agreements at EU level has also remained very low. By the early 2000s, only 12 EU-wide environmental voluntary agreements have been adopted despite considerable efforts by the EU Commission's to increase their uptake (CEC 2002). EU-wide voluntary agreements are normally negotiated between the Commission and companies. Once agreement has been reached between these two actors they are communicated to the Council and European Parliament (EP) the latter of which has traditionally been hostile towards the adoption of EU-wide voluntary agreements for legitimacy and transparency reasons which helps to explain their low uptake. EU-wide voluntary agreements have to be adopted outside the formal decision-making procedures as they are not listed in the EU Treaties as possible policy instruments for the EU. The EP therefore lacks a clearly defined role in the adoption process of EU-wide voluntary agreements which weakens the legitimacy of this type of policy instrument on the EU level. Moreover, as voluntary agreements cannot be enforced under the EU Treaties, there are serious concerns about free-riding which is also a potential weakness of member state voluntary agreements. However, these concerns are magnified on the EU level because a much larger number of actors are usually involved thus making monitoring more challenging while in terms of competition more is at stake in the much larger Single European Market (SEM) than member states' national markets.

4.3 Usage Patterns of Market-based Instruments

4.3.1 Eco-taxes

Japan and Germany adopted some of the world's first eco-taxes in the early 1970s (Andersen 1996; Andersen and Sprenger 2000). However, although Germany adopted a wastewater levy already in 1974 it was actually not fully implemented until the early 1980s (e.g. Wurzel, Zito and Jordan 2013). A more systematic use of eco-taxes occurred in Europe first in the Netherlands

and the Nordic countries in the 1980s and 1990s (Andersen 1996; Andersen and Sprenger 2000: 27). The UK adopted major national eco-taxes only from the 1990s onwards (OECD 1993).

There is wide variation in the type of eco-taxes adopted in the five jurisdictions assessed in this chapter. Between the early 1990s and early 2000s, the Netherlands adopted the most comprehensive range of eco-taxes. Germany introduced a major ecological tax reform in 1998 and a raft of additional eco-taxes in 2010. In other words, Germany developed further this particular policy instrument in the late 1990s. Austria and the UK have adopted more modest eco-taxes in a narrower range of sectors.

The Dutch eco-taxes evolved from environmental charges and levies which were used to support specific regulatory efforts to more encompassing general eco-taxes, German eco-taxes evolved more gradually and very much in an ad hoc fashion until the adoption of the ecological tax reform in 1998. Austria set up a Tax Reform Commission which reported on a possible ecological tax reform which, however, was not adopted. Instead the Austrian government continued to fine tune existing eco-taxes while incrementally adding new ones. The UK has innovated with hypothecated eco-taxes and a wide range of eco-taxes in the 1990s but has since fallen behind again. Germany, the Netherlands and UK innovated with eco-taxes that were explicitly linked to a reduction in labour costs (e.g. in the form of reduced national insurance and/or pension contributions). Although they tend to emit a large number of greenhouse gas emissions, high energy users which are in direct competition with companies abroad have been granted generous exemptions from eco-taxes in Austria, Germany, the Netherlands and UK. The EU has so far failed to adopt EU-wide eco-taxes despite the European Commission's strong push for an EU-wide carbon dioxide/energy tax in the early 1990s. The main reason for this is the unanimity

requirement for the adoption of all taxes on the supranational EU level against which the UK has been opposed on sovereignty grounds.

4.3.2 Emissions trading

At the insistence of the USA, the 1997 Kyoto Protocol stipulated emissions trading (and other flexible instruments such as Joint Implementation (JI) and the Clean Development Mechanism (CDM) as a possible policy instrument for achieving the greenhouse gas reductions. The EU and most of its member states were initially strongly opposed to emissions trading although they eventually gave in to American pressure (Grubb, Vrolijk and Brack 1999; Wurzel 2008). Only the UK (and Denmark) as well as, although to a much lesser degree, the Netherlands experimented with national emissions schemes prior to the setting up of the EU ETS.

The EU ETS, which distinguished a first trading phase (2005-07) and second trading phase (2008-12), was initially a highly decentralised ETS for CO₂ emissions from stationary sources. A third trading phase (2013-2020) was added during the revision of the EU ETS which centralised the rules of the scheme. The Commission acted as an emissions trading policy instrument entrepreneur. Frustrated by the lack of progress of its carbon dioxide/energy tax proposal, which had been vetoed by the UK, and concerned about competing national ETSs, the Commission proposed an EU-wide emissions trading in a Green Paper in 2000. It was strongly supported in its efforts by the majority of member states (including the UK and the Netherlands) and the EP. The German government remained opposed to emissions trading until well into the adoption phase of the EU ETS. Only after the 2002 national elections in which the Greens gained a significant number of seats in parliament the German government started to support the adoption of the EU ETS.

Austria and Germany initially acted as emissions trading laggards. The German Chancellor Gerhard Schröder almost torpedoed the adoption of the EU ETS. However, Austria and Germany (as well as the Netherlands and UK) have since supported the tightening and centralisation of the rules of the EU ETS. In the second and third trading phases Germany has made wide use of auctioning while Austria recognised that the EU ETS had become an essential policy instrument for achieving its Kyoto Protocol greenhouse gases reduction target.

4.4 Environmental Regulations

Gunningham and Grabosky (1998: 38) have argued that ‘[t]he dominant government response [to the arrival of environmental issues on the political agenda], ... has been the application of “direct” or “command and control” regulation designed to prohibit or restrict environmentally harmful activities’. Regulation remains important even in jurisdictions where NEPI use is high, such as in the Netherlands and Germany. In short, there has not been a wholesale switch to NEPIs. Instead NEPIs seem to have *supplemented* rather than *supplanted* regulatory tools (see also Mol, Lauber and Liefferink 2000; Holzinger and Knill 2003; Wurzel, Zito and Jordan 2013; Interviews, Member State official, 2017). NEPIs often ‘fill in the cracks’ not covered by regulation or deal with emerging issues like climate change which are difficult to tackle with regulations. Moreover, many NEPIs also require regulations, for example, to set the rules for their operation. Gunningham and Grabosky (1998) have therefore argued that hybrid instruments which are made up of both NEPIs and regulations have emerged.

Importantly, regulation remains the mainstay of EU environmental policy despite the fact that a reduction in their adoption rate has taken place in the 2010s. The EU has adopted a relatively large number of traditional environmental regulations (in the form of directive, regulations and

decisions). However, it has struggled greatly to develop a popular eco-label scheme, been very slow in adopting voluntary agreements, and has failed to clear the unanimity requirement (among member states) for EU-wide eco-taxes. A strong EU entrepreneurial NEPIs influence is discernible only with respect to emissions trading. Relatively few EU level NEPIs exist despite political commitments to adopt more of them (see the Commission's 2001 White Paper on *European Governance* (CEC 2001)). One important reason for this is that member states tend to support the adoption of common policy objectives at the EU and/or international level but are keen to reserve the right to determine the instruments of achieving them.

Environmental regulation has taken on an important supporting role for the adoption of many NEPIs. This is the case in particular for market-based policy instruments such as emissions trading and eco-taxes. Gunningham and Grabosky (1998: 391) have pointed out that '[s]ome economic instruments, such as taxes and charges, are high on coercion and low on prescription'. It was arguably only due to the 'shadow of the law' (Héritier and Lehmkuhl 2008) that a large number of voluntary agreements were adopted in Germany and, although to somewhat lesser degree, in Austria in the 1990s. The Dutch *convenants* constitute quasi-legal agreements. Many NEPIs therefore rely on regulation.

A cross-jurisdictional trend towards greater flexibility in environmental regulations is discernible across all five jurisdictions where traditional environmental regulations have become less rigid and more flexible. Since the early 1990s, EU and member state environmental regulations have become more flexible and innovative. In other words, European environmental regulations have become 'smarter' (Gunningham and Grabovsky 1998). While remaining dominant in most jurisdictions regulation has evolved over time. It is often adopted in 'smarter' or more 'light handed' forms when compared to the early 'command-and-control' regulations

which featured large in the 1970s (Gunningham and Grabosky 1998; Rengeling and Hof 2001). For example, in the UK, the national integrated pollution control regime (i.e. regulation) shares many similarities with what continental states describe as negotiated agreements underpinned by the law (e.g. the Dutch covenants).

Gunningham and Grabosky's (1998) use 'smart regulation' as an umbrella term which subsumes a wide range of NEPIs including eco-labels, environmental management systems, voluntary agreements, eco-taxes and ETSs. In this chapter, we distinguished conceptually between environmental regulations and NEPIs although we also paid attention to the hybridisation of 'old' instruments (i.e. regulations) and 'new' tools (i.e. NEPIs). This has helped us to identify the changing patterns of the use of traditional regulation and NEPIs in four member states (Austria, Germany, the Netherlands and UK) and on the EU level as well as the changing policy instrument mixes which they have produced.

5. Leaders, Followers and Laggards

Table 4.2 summarises the adoption patterns for NEPIs in the five European jurisdictions assessed in this chapter. Importantly the analytical descriptors - leader, follower and laggard - used in Table 4.2 (like in Table 4.1 above) offer a relative assessment between the five jurisdictions listed, rather than an absolute ranking. NEPI leaders as defined in Table 4.2 are jurisdictions which have first made significant use of a particular NEPI type. Followers eventually catch up with the leaders while laggards either fail to adopt a particular type of NEPI or utilise it only in an insignificant manner.

Table 4.2: Leaders, followers and laggards

Types of NEPIs	Leader	Follower	Laggard
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Informational instruments	Eco-label	Germany	Austria, EU Netherlands	UK
	EMS/EMAS	UK, EU	Germany, Austria	Netherlands
Voluntary instruments	Voluntary Agreements	Germany, Netherlands	Austria, UK	EU
Market-based instruments	Eco-taxes	Netherlands, Germany	Austria, UK	EU
	Emissions trading	UK, EU, Netherlands	Austria, Germany	---

Source: Adapted from Wurzel, Zito and Jordan (2013) with updates.

Leaders may eventually be overtaken by followers in the use of a particular NEPI. In order to achieve a better domestic fit and/or to improve its effectiveness, followers often modify NEPIs which they have transferred voluntarily from another jurisdiction or which has been ‘coercively’ transferred to their jurisdiction. Germany acted as an early leader for eco-label schemes when it set up its domestic Blue Angel scheme in 1978. However, it was primarily the general idea behind this NEPI type which was transferred to other jurisdictions (including the EU) rather than the specific rules of the original German eco-label scheme. Austria (1990), the Netherlands (1992) and EU (1992) all followed the German example only after a considerable time lag but without simply copying the German Blue Angel scheme. The followers all adopted domestic eco-label schemes which exhibited some novel features that reflected both domestic priorities and the desire to achieve a good domestic fit and increased effectiveness.

The UK set the pace for EMSs with its innovative BS 7750 voluntary standard which also influenced the EU's EMAS. Germany and Austria initially opposed the EU's proposal for EMAS. However, once the EU had adopted EMAS, the followers Germany and Austria quickly became the highest EMAS users leaving well behind the UK in EMAS registrations. Germany and Austria lobbied hard both to make EMAS more ambitious and to extend its scope from private companies to public institutions such as government ministries and agencies. The Netherlands never developed a national EMS and has made only little use of EMAS. Dutch firms have instead preferred the ISO 14001 standard which is environmentally less ambitious but has a global reach.

The Netherlands and Germany have been early innovators and heavy users of voluntary agreements. However, the core features of Dutch and German voluntary agreements differ significantly (e.g. Wurzel, Zito and Jordan, 2013). The Netherlands makes wide use of sector-wide covenants which are legally binding and monitored by independent institutions. German voluntary agreements are non-binding agreements (*Selbstverpflichtungen* i.e. self-binding agreements) usually put forward by companies with the aim of pre-empting government regulation. Only a small number of German voluntary agreements has been monitored by an independent verifier. Austria became a voluntary agreements follower in the 1980s when it partly emulated the non-binding German voluntary agreements (rather than the Dutch covenants). However, after extensive usage in the 1990s there was a steep decline in voluntary environmental agreements usage in Austria in the 2000s. Constitutional reasons have prevented Austria and Germany from adopting legally binding voluntary agreements such as the Dutch *covenants*. The UK has adopted only a moderate number of voluntary agreements. The lack of a clear Treaty base for voluntary agreements prevented EU Commission officials from trying to emulate the Dutch covenants (Interviews, 2001 and 2004). Moreover, concerns about free riders are

magnified at the EU level due to the significantly large number of actors involved and the large size of the SEM which makes more important arguments about fair competition.

Importantly, although the early leaders (Germany and the Netherlands) have remained the largest users of voluntary agreements within the EU, their popularity declined even in the two innovator states in the 2000s and even more so in the 2010s (see Table 4.2). One important explanation for the declining popularity in the use of voluntary agreements is that they have been supplanted by other NEPIs and/or traditional environmental regulations. For example, the EU ETS rendered obsolete some Dutch, German and Austrian voluntary agreements aimed at reducing greenhouse gas emissions while legislation has overwritten the EU-wide voluntary agreement on the reduction of CO₂ emissions by the car industry.

The Netherlands was an early eco-taxes innovator which has consistently made use of a wide range of eco-taxes. Germany was also an early eco-taxes pioneer when it adopted the 1976 waste water levy (Andersen 1994). However, although Germany strongly supported the adoption of an EU-wide carbon dioxide/energy tax, its domestic use of eco-taxes remained moderate until the adoption of an ecological tax reform in 1998. Austria and the UK were followers which made use of eco-taxes only belatedly when compared to the Netherlands and Germany. However, the UK adopted some innovative eco-taxes (e.g. the fuel escalator) which influenced the eco-tax thinking of environmental policy makers in Austria, Germany and the Netherlands. The EU is an eco-taxes laggard because it failed to adopt a supranational eco-tax. One important reason for this is because the UK has consistently vetoed taxes on the EU level on sovereignty grounds. Whether the UK's exit from the UK may pave the way for the adoption of EU-wide eco-taxes remains however doubtful. Member states other than the UK (e.g. traditionally Spain and more recently the Visegrad countries) have traditionally also been opposed to EU-wide eco-taxes for

fear that this may put them at an economic disadvantage. Because the EU failed to adopt a supranational carbon dioxide/energy taxes, a group of like-minded European countries met between 1994 and 1998 to discuss the practical experience with and possible transnational coordination of national eco-taxes although little progress was made.

The UK became an emissions trading innovator when it set up Europe's first domestic ETS for greenhouse gases in 2002. Up to the early 2000s, the Netherlands experimented only with rudimentary small-scale domestic ETSs (e.g. for manure). In 2003 the EU adopted the world's first supranational ETS for CO₂ emissions. Austria and Germany which initially acted as reluctant ETS followers eventually became supportive of a relatively ambitious EU ETS.

Importantly, emissions trading constitutes the only NEPI type assessed in this chapter for which the EU has acted as a genuine innovator. However, the EU was only a reluctant emissions trading pioneer (Wurzel 2008). It is unlikely that the EU ETS and UK ETS 'could have been adopted as quickly as they were without America setting a domestic example and insisting on emissions trading in the 1997 Kyoto Protocol' (Wurzel 2008: 5).

Germany (eco-labels, voluntary agreements and, although to a lesser degree, eco-taxes) and the Netherlands (voluntary agreements, eco-taxes and, although to a lesser degree, emissions trading) have both acted as leaders for three types of NEPIs albeit different ones. The UK (EMS and emissions trading) for two types of NEPIs and the EU (emissions trading) were leaders for only one NEPI type. Austria is the only jurisdiction which failed to become a leader for any of the five NEPIs assessed in this chapter. Nevertheless, Austria always followed the NEPIs leader thus avoiding the status of NEPIs laggard. Similarly, Germany always acted as a follower in cases where it was not a NEPIs innovator. The UK (eco-labels)

and EU (voluntary agreements and eco-taxes) acted as laggards for one and two NEPI types respectively. The EU had adopted only about a dozen EU-wide voluntary agreements by the 2010s; this amounts to an insignificant use of voluntary agreements when compared to the large number of voluntary agreements adopted by especially the Netherlands and Germany. The Netherlands could be classified as an EMS laggard because it made only insignificant use of EMAS and failed to adopt a domestic EMS. However, Dutch firms have made relatively wide use of the ISO 14001.

The analytical classification of leader, follower and laggard can mask significant changes in the use of particular NEPIs that may take place in a certain jurisdiction over time. Table 4.2 illustrates the changing usage patterns for the different types of NEPIs in each of the five jurisdictions. It shows that once a certain type of NEPI has been adopted by a particular jurisdiction its usage tends to change only gradually. The usage of voluntary agreements seems to have peaked in the Netherlands, Germany and Austria in the 1990s. Similarly the drive for eco-taxes lost momentum in the 2000s. The EU ETS, which became operational in 2005, is the only NEPI whose usage expanded significantly in the 2000s (and early 2010s). These findings provide empirical evidence for the claim that ‘re-tooling’ processes take place over time (Hood 1983: 126).

This contribution has assessed the role of the EU in terms of the ability to innovate ‘old’ and ‘new’ policy instruments. However, there is a broader question about what impact the supranational EU has had on the member state policy instrument mixes. Because the EU has often been the NEPIs laggard its role in transforming the policy mix of the member states has largely focused on climate change and particularly the ETS, which involved all member states having to reshape their climate governance approach to some extent. With the exception of the ETS, the EU has not added particularly to the member states policy

instrument mixes with the important exception of traditional regulation. This is not to downplay the EU role as, from the 1970s, the EU has issued a relatively large number of supranational environmental laws creating common standards and objectives that its member states have had to follow. Over time, those regulations have generated their own innovations. Accordingly the Water Framework Directive was notable for creating certain processes which enhanced dialogue with society (Page and Kaika 2003). This harkens back to the notion of ‘smart regulation’, where the instrument contains several different mechanisms (Gunningham et al. 1989). By contrast the regulatory framework that protects the operation of the SEM means that the EU, in the form of state aid and competition policy concerns, has had a prohibitive or at least limiting impact on member states seeking to expand their mixes with, for example, subsidies and other incentives. For example, the German ecological tax reform could be adopted by the German parliament only after it had been altered to comply with instructions from the European Commission (Wurzel, Zito and Jordan 2013).

CONCLUSION

The chapter assessed how policy instrument mixes appear in the real world as opposed to the theoretical world of (especially economic) textbooks. It has shown how the mixes have changed over a period of almost five decades by focusing on traditional regulation and three different types of NEPIs - informational instruments, voluntary agreements and market-based instruments (eco-taxes and emissions trading) in five jurisdictions, namely Austria, Germany, the Netherlands and UK as well as the EU. The empirical focus on NEPIs and traditional regulation allowed us to analyse whether new modes of *governance* have supplanted or merely supplemented traditional methods of *government* (i.e. command-and-control regulation) in different European jurisdictions. Clearly there has been a significant uptake of NEPIs in all five

jurisdictions. However, important differences remain between the specific policy instrument mixes across different jurisdictions but also within one and the same jurisdiction over time. A certain degree of hybridisation of ‘old’ and ‘new’ instruments towards what Gunningham and Grabosky (1998) have termed ‘smart regulation’.

Seeking to specify what is an optimal mix a priori is an important and interesting theoretical topic, but it does not really explain why instruments are actually mixed in the real world of policy and politics. Taking a longitudinal empirical perspective has allowed us to identify leaders, followers and laggards for different types of NEPIs. It has revealed policy instrument diffusion patterns that are influenced by jurisdictional path-dependencies that can lead to very different sequencing of the adoption of certain types of NEPIs. For example, Germany adopted a national eco-label scheme already in 1978 while Austria (1990), the Netherlands (1992) and EU (1992) followed the German example only after a considerable time lag. By the late 2010s the UK still had not adopted a national eco-label scheme while relying instead on the EU eco-label at least until it exists from the EU.

In all five jurisdictions, complex policy instrument mixes have emerged which blend the ‘old’ and the ‘new’ in puzzlingly different ways. In the language of the (new modes of) governance literature what we seem to be witnessing is arguably best described as ‘governance-cum-government’. However, there is little empirical evidence for the widely held claim that non-coercive policy instruments (such as informational instruments and voluntary agreements) should be used before coercive instruments are adopted. This has created a complex mixture of (policy instrument) continuity and change in different jurisdictions which may use differently one and the same type of policy instrument.

Classic administrative theory largely considered policy instrument selection as ‘a neutral, even scientific exercise’ (Bemelmans-Videc, Rist and Vedung 1997: 268). However, Hood (1983: 136) has flagged up that:

Given that all feasible alternatives cannot be systematically approached, it follows that in many cases politics will play a large part in the selection of tools for the job... Indeed, very commonly it is the instrument selected for reaching a policy aim that is far more contentious than the aim itself.

The jurisdictional context variable (e.g. national or sectoral policy styles) play an important role for both the selection and specific design of policy instruments.

Once established within a particular jurisdiction, policy instruments tend to retain their relative role in the policy instrument mix for a considerable period of time. However, as Smith and Ingram (2002: 598) have pointed out: in the real world of politics and power, ‘the tool box is not constant. It changes over time as a result of the invention of new tools, national and international developments, and the interactions of the supranational institutions’. Bridging the theoretical accounts, which are often informed by economic cost-effectiveness considerations, with more political and institutional approaches of the evolution of policy instrument mixes in the real world should help us to understand better why it is often difficult to move from ‘not so smart policy instrument mixes’ towards smart policy instrument mixes.

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